

## Title Page

### Plant community compositional stability over 40 years in a Fraser River Estuary tidal freshwater marsh

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**Commented [MT(1):** What about Mary – has she had an opportunity to contribute?

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#### Acknowledgements

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## Abstract

*Wetlands: Please provide an abstract of 150 to 250 words. The abstract should not contain any undefined abbreviations or unspecified references.*

Long-term data sets documenting temporal changes in vegetation communities are uncommon, yet imperative for understanding trends and triggering potential conservation management interventions. For example, decreasing species diversity and increasing exotic species abundance may be indicative of decreasing community stability. We explore long-term plant community change over a 40-year period through the contribution of data collected in 2019 to two historical datasets collected in 1979 and 1999 to evaluate decadal changes in plant community biodiversity in a tidal freshwater marsh in the Fraser River Estuary in British Columbia, Canada. We examine whether characteristic plant assemblages are consistent over time, whether alpha ( $\alpha$ ) and beta ( $\beta$ ) diversity change within and between assemblages, and whether associated indicator species changed. We found that while plant assemblages were characterized by the same dominant indicator species while most other indicator species changed, and that  $\alpha$ -diversity decreased while  $\beta$ -diversity increased. Further, we found evidence for plant assemblage homogenization through the increased abundance of exotic species. These observations may inform concepts of habitat stability in the absence of pulse disturbance pressures, and corroborate globally observed trends of native species loss and exotic species encroachment. Our results indicate that within the Fraser River Estuary, active threat management may be necessary in areas of conservation concern in order to prevent further native species biodiversity loss.

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## Keywords

*Wetlands: four to six*

shifting baselines; reference conditions; dispersal networks; species turnover; conservation land management

## 1 Introduction

2 In a time of rapid global change, temporal shifts in plant community composition can indicate ecosystem  
 3 stress response and inform conservation management interventions. Shifts in community-dominant species, loss  
 4 of native species diversity, and increasing abundance of exotic species may indicate loss of stability through loss  
 5 of species or functional redundancy (Donohue et al., 2016; Holling, 1973). In turn, this may indicate reduced  
 6 resistance to change or capacity to recover from disturbance, known as resilience (Bai, et al., 2004; Tilman,  
 7 Reich, & Knops, 2006). Furthermore, the local loss of native species may have stronger negative impacts on  
 8 regional biodiversity persistence when the regional pool of potential species is reduced or environmentally  
 9 constrained (Hanski, 1982; Lepš, 2004). Characterization of plant community changes on decadal timescales  
 10 contributes to observation of meaningful long-term patterns of compositional stability, and is instructive for  
 11 developing hypotheses to test drivers of disturbance, especially in dynamic landscapes heavily impacted by  
 12 anthropogenic activities such as estuaries (Ovaskainen et al., 2019; Underwood et al., 2000).

13 Estuaries are at the terrestrial-marine interface where hydrogeomorphic and ecological changes occur  
 14 on annual, decadal, and millennial timescales (Pasternack, 2009). Estuarine habitats support high species  
 15 richness, including species at risk (Kehoe et al., 2021) and are important carbon reservoirs (Douglas, et al., 2022).  
 16 Because these ecosystems will experience accelerated change under sea level rise, they are of increasing  
 17 conservation concern (Brophy et al., 2019); understanding estuarine habitat stability can inform global change  
 18 resilience strategies. Estuaries in North America are of particular conservation importance in the Pacific  
 19 Northwest (PNW) because their pathways of retreat or expansion are often spatially restricted by fjord  
 20 geography (Emmett et al., 2000), whereas estuaries along the Atlantic coast may spread along expansive coastal  
 21 plains. Tidal freshwater marshes are the upper reaches of estuaries dominated by riverine freshwater, and in the  
 22 PNW they are particularly important as early transitional habitat along a salinity gradient for anadromous  
 23 salmonids (Chalifour et al., 2019; Davis et al., 2021). Within the Fraser River Estuary 85% of floodplain and 64%  
 24 of stream habitat has been lost (Finn et al., 2021), emphasizing the need to understand the condition of  
 25 remaining estuarine habitat. Estuary conservation efforts are intended to protect coastal municipalities and  
 26 provide sufficient habitat for wildlife, and stability of plant communities within tidal marshes contribute to the  
 27 ability of these habitats to resist change or recover from disturbance (Holling, 1973).

28 A challenge of understanding community stability, including within estuaries, is the lack of long-term  
 29 data. In the absence of long-term monitoring, historical datasets can provide a ‘snapshot’ of species  
 30 compositional variation over time. One such opportunity exists in the Fraser River estuary, British Columbia,  
 31 Canada in an area called Ladner Marsh ([Figure 1](#)). Despite large-scale industrialization and urbanization within  
 32 the region, Ladner Marsh has escaped development, and to the best of our knowledge has not experienced  
 33 major natural or anthropogenic disturbance in the past 50 years. Two historical studies conducted in Ladner  
 34 Marsh (Bradfield & Porter, 1982; Denoth & Myers, 2007) used similar methods to document floristic diversity.  
 35 While these studies independently characterize different community metrics, these datasets provide the  
 36 opportunity to repeat observations and characterize long-term plant community changes and habitat stability.  
 37 We used three observational datasets spanning four decades to answer the following questions:

- 38 (1) Are tidal freshwater marsh assemblages characterized by the same dominant species over a 40-year period?  
 39 In the absence of significant environmental disturbance, we expect the same species composition to  
 40 dominate each assemblage as identified by Bradfield & Porter (1982).  
 41 (2) Are assemblages characterized by similar indicator species? If not, which species gained or lost are driving  
 42 changes within each assemblage? We expect that increasing abundance of invasive species over time would  
 43 result in greater net number of native species lost (and fewer net native species gained).

These changes  
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 disturbance.  
 See Bruno 2000  
 Ecology, for  
 example.  
 So I think this  
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 as an alternative  
 to null hyp.

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45 (3) Is the mean species diversity ( $\alpha$ -diversity) and variation ( $\beta$ -diversity) within and across assemblages constant  
 46 between the three sampling periods (1979, 1999, 2019)? If the plant community is stable, we expect little  
 47 change in  $\alpha$ -diversity and  $\beta$ -diversity.

## 48 Methods

### 49 Physical & ecological context

50 The Fraser River is the largest catchment in British Columbia, covering one quarter of the province. The estuary  
 51 currently spans 2,814 ha, one-third of which lies within the South Arm Marshes Wildlife Management Area  
 52 (Schaefer, 2004) (Figure 1). Ladner Marsh occupies approximately 100 ha within the South Arm Marshes,  
 53 bounded to the east by urban and industrial development and to the west by the Fraser River (Figure 1).

54 Plant species common to these habitats are generally herbaceous, and the community is largely  
 55 dominated by sedges and rushes with some salinity tolerance, but a greater diversity of herbaceous flowering  
 56 species (hereafter, forbs). Forb species such as bogbean (*Menyanthes trifoliata* L.) are tolerant of continuously  
 57 waterlogged conditions, whereas sedges (*Carex lyngbyei* Hornem.) are better adapted to microsites that are  
 58 regularly inundated and drained (Bradfield & Porter, 1982). Grass species such as non-native tall fescue (*Festuca*  
 59 *arundinacea* Schreb.) may prefer the most well-drained sites, although some non-native species such as reed  
 60 canary grass (*Phalaris arundinacea* L.) tolerate more saturated soils, and present an invasion threat in tidal  
 61 wetlands (Sinks et al., 2021).

62 In Ladner Marsh, Bradfield & Porter (1982) identified distinct community sub-types (hereafter,  
 63 "assemblages"). Denoth & Myers (2007) repeated the sampling to determine whether an exotic species was  
 64 displacing a listed species of conservation concern in British Columbia, Henderson's checker-mallow (*Sidalcea*  
 65 *hendersonii* S. Watson). Henderson's checker-mallow is locally abundant in this marsh, and thus stability of  
 66 sufficient habitat is vital for its conservation.

### 67 Vegetation surveys

#### 68 1979 and 1999

69 The first data collection was conducted in 1979 (Bradfield & Porter, 1982). Eight transects were  
 70 positioned along a north-to-south gradient, and 1 m<sup>2</sup> quadrats (plots) were placed in the center of vegetation  
 71 patches where species composition noticeably changed, or every 10 m along the transect, whichever distance  
 72 was shorter (Bradfield, 2019 personal comm.) (Figure 1D). Cluster analysis and principal components analysis  
 73 (PCA) distinguished three assemblages, each dominated by a distinct species: Sedge (*Carex lyngbyei*), Fescue  
 74 (*Festuca arundinacea*), and Bogbean (*Menyanthes trifoliata*). Bradfield & Porter (1982) hypothesized that  
 75 edaphic factors drove assemblages, such as waterlogged soils in the bogbean assemblage, or drainage along  
 76 channel edges in the fescue assemblage.

77 A subsequent survey conducted in 1999 recreated the transects and sought to place sampling plots at  
 78 the exact positions sampled in 1979 to test relationships between invasive purple loosestrife (*Lythrum salicaria*,  
 79 L.) and Henderson's checker-mallow (*Sidalcea hendersonii*) (Denoth & Myers, 2007). While Denoth & Myers did  
 80 not seek to test changes in community composition, data were collected according to the same protocols as in  
 81 1979, and the data have been made available for comparison. This publication will reference dates the data  
 82 were collected, rather than publication dates of the corresponding studies.

#### 83 2019

84 No permanent markers were left in Ladner Marsh, so precise transects assessed by Bradfield & Porter  
 85 (1982) or Denoth & Myers (2007) were not identifiable in 2019. Transect endpoints were approximated within  
 86 ~5 m by overlaying Figure 1 in Bradfield & Porter's 1982 publication (Figure 1D) on a georeferenced base map,

State the goal: We aimed to resurvey  
 the exact locations as much as possible.

OR: We aimed to sample the marsh in a representative  
 way to allow comparison of the 1979 and 1999  
 samples.

This is more like (what's)  
 known methods is what exactly  
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 hypothesized?  
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 why it's in  
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methods needs more numbers & detail  
So it's clear how data & uncertainties  
Can be compared across studies.  
And how many samples, indiv, sp, etc

presumably, dependent  
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on replication,  
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bias if sample  
size varied  
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big range, mention  
potential for seasonal  
effects in discuss.

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that these methods  
work w/ the data.

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This description  
suggests individuals  
were not systematically  
collected and identified.

I think these statistics  
assume individuals - common  
AND rare, an identified  
and counted.

clarify above how  
individuals were counted. Were  
all stems counted and ID'd?  
or random subset (~50%)?

Lane et al., 2022. Plant community stability, 6

131

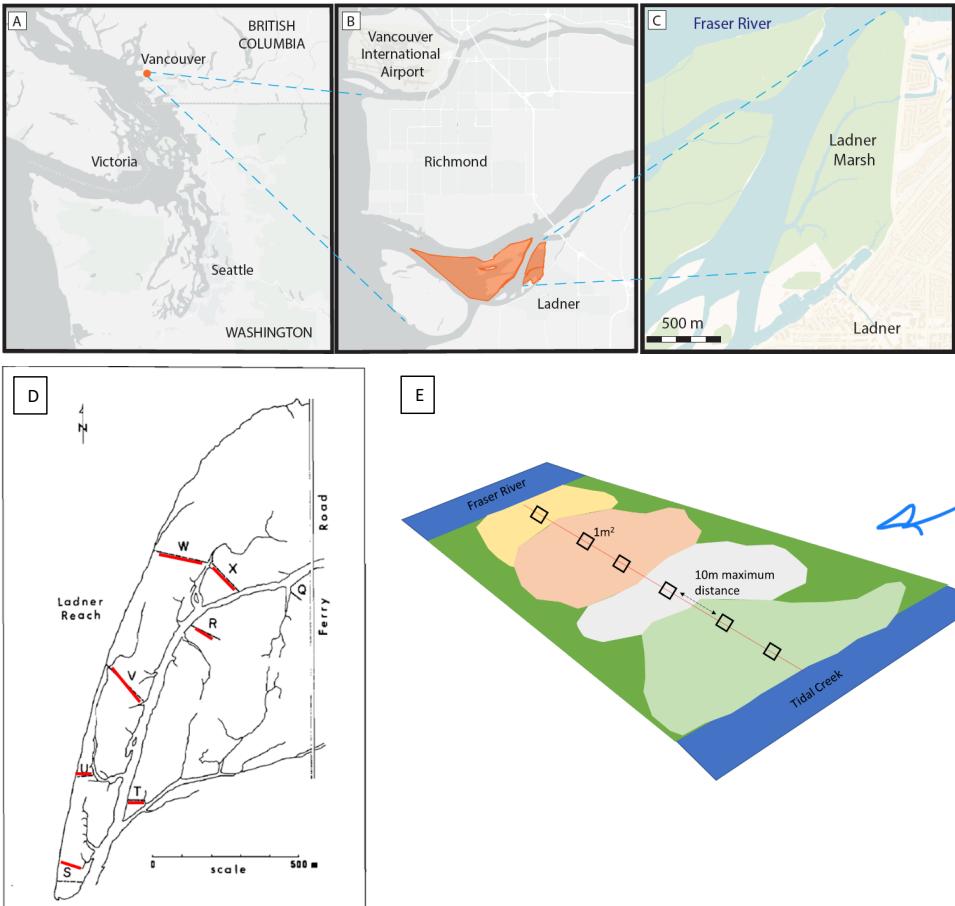
132 Community diversity calculations followed Whittaker (1975), with  $\alpha$ -diversity calculated as the mean  
133 number of species per quadrat within an observation year and assemblage, and  $\beta$ -diversity calculated as the  
134 total number of species within the assemblage divided by  $\alpha$ -diversity. These calculations were also performed  
135 on all data recorded for the observation year to generate a community-wide measure of diversity. Community  
136 turnover for each assemblage was measured using the "codyn" R package (Hallett et al., 2016). Total species  
137 turnover (total magnitude of change), species gained (appearances), and species lost (disappearances) were  
138 calculated as a percent change for each assemblage between 1979-1999, and 1999-2019. Total turnover was  
139 calculated as a ratio of the absolute value of species gained and lost to the total number of species observed in  
140 both timepoints.

141 During analyses, both Euclidean and Bray-Curtis distances were used to confirm distance measure did  
142 not have a major effect on plot clustering and subsequent indicator species analysis; cluster analysis figures and  
143 indicator species table using Bray-Curtis distance are available in [Table 4](#) and [Figure 4](#). To address inconsistent  
144 numbers of plots grouped into assemblages each year, diversity metrics were bootstrapped 10 times using the  
145 minimum number of plots observed in an assemblage each year ( $n = 18$ ) ([Table 5](#)). All analyses were performed  
146 in R v.4.0.2 (cite the R team).

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153 **Figure 1.** Location of the study site in the Fraser River Estuary, Vancouver, British Columbia, Canada (A), approximately 20 km north of the  
 154 South Arm Marshes Wildlife Management Area (highlighted in orange). B). Ladner Marsh abuts municipal development on the south bank  
 155 of the Fraser River (C). Base maps (A, B) generated by iMap published by the B. C. Conservation Data Center (Victoria, BC, Canada,  
 156 <https://maps.gov.bc.ca/ess/hm/imap4m/>) and (C) OpenStreetMap (OpenStreetMap contributors, 2015,  
 157 <https://www.openstreetmap.org/>) (Lane, 2022). (D) Transect locations illustrated in 1982 publication figure (line drawing), which was  
 158 overlaid on Google Earth basemap to relocate transects in 2019 (red), and (E) semi-systematic plot placement within and between  
 159 assemblages based on species dominance.

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handwritten note: nice figure, v. helpful

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## 161 Results

162 Three main assemblages within Ladner Marsh can be characterized by the same dominant indicator  
 163 species across all sampling periods: Sedge (*Carex lyngbyei*), Fescue (*Festuca arundinaceae*), and Bogbean  
 164 (*Menyanthes trifoliata*). While the three assemblage indicator species remain constant over time and drive  
 165 cluster groups, other species that significantly drive indicators of assemblages change over time (Table 2)  
 166 **Error! Reference source not found.**) For example, in 1979 the indicator species defining the Sedge assemblage cluster  
 167 were *C. lyngbyei*, *Sagittaria latifolia* Wild., and *Schoenoplectus tabernaemontani* (C.C.Gmel.) Palla, however in  
 168 1999 the same assemblage included indicator species *C. lyngbyei*, and *Impatiens capensis* Meerb. By 2019, *C.  
 169 lyngbyei* was the only indicator for this assemblage. Similarly, *F. arundinaceae* remained a common indicator  
 170 species within the Fescue assemblage, but the assemblage lost four out of seven total indicator species between  
 171 1979-2019. While the identities of the remaining indicator species changed, there was no strong trend of  
 172 changes in clade, or potential difference for changes in ecological function based on qualitative review of  
 173 changing species identity. ↓

174 Across the entire Ladner Marsh plant community, two to three species were lost each year following the  
 175 1979 survey. Within every assemblage  $\alpha$ -diversity (mean number of species per plot) decreased every  
 176 observation year, while  $\beta$ -diversity (ratio of total species in the assemblage to  $\alpha$ -diversity) increased each year  
 177 for all assemblages (Table 1). For example, the Sedge community suffered the least loss of species and  $\alpha$ -  
 178 diversity across sampling years, although  $\beta$ -diversity increased as in other assemblages, indicating increasing  
 179 variability (and thus increased rarity) in which species may be encountered within a given assemblage. The  
 180 Fescue assemblage had the greatest loss of  $\alpha$ -diversity (> 50%) between 1979 and 2019. Nearly 50% fewer plots  
 181 clustered as Fescue in 2019 than in 1979, however bootstrapping 18 random plots from every sampling year  
 182 showed the same trend, indicating that loss of species was not related to loss of plots (Table S5). Total  
 183 magnitude of species turnover between 1999 and 2019 was ~50% in each assemblage, largely driven by greater  
 184 species disappearance (loss) between 1999 and 2019 (Table 6).

185 The greatest loss of native species richness occurred in the Fescue assemblage, while gains in exotic  
 186 richness were found in all assemblages (Table 7). The Fescue assemblage had a net loss of 18 native species  
 187 between 1979 and 2019. Among the species lost from the Fescue assemblage, 12 were lost from all three  
 188 assemblages (six forbs, six graminoids), or were never found in any other assemblage. Species gained include  
 189 two woody species, and one each of forb, graminoid, and fern ally (*Equisetum arvense* L.). There was a net zero  
 190 gain of exotic species in the Fescue assemblage, however exotic *Phalaris arundinaceae* (reed canary grass)  
 191 accounts for the greatest 2019 mean cover in the entire assemblage (25-50% mean cover). In the Bogbean  
 192 assemblage, new exotic species include *P. arundinaceae* and *Iris pseudacorus* (yellow flag iris). Within the Sedge  
 193 assemblage, there was a net loss of 3 native species, and net gain of 3 exotic species, including *P. arundinaceae*  
 194 and *I. pseudacorus*. As of 2019, these species accounted for  $\leq 25\%$  mean cover, but may be of significant  
 195 management concern.

196 Cover abundance of species significantly defining assemblage associations show an overall trend of  
 197 decreasing cover over time (Figure 3). Notably, Fescue assemblage shows ~50% decrease in cover of its exotic  
 198 indicator *F. arundinaceae* between 1979 and 2019, while cover of exotic *P. arundinaceae* tripled since 1999. In  
 199 the Sedge assemblage both native indicator sedge *C. lyngbyei* and exotic indicator grass *A. stolonifera* had  
 200 decreased cover abundance from 1979-2019 (Figure 2, **Error! Reference source not found.**), with each species  
 201 losing ~25-35% cover abundance between 1979-2019. Meanwhile, exotic species *Lythrum salicaria* and *F.  
 202 arundinaceae* increased ~50% and 100%, respectively, in abundance ( $\leq 25\%$  mean cover) by 2019 (Table 7).  
 203 Similarly, in the Bogbean assemblage, cover abundance of native keystone species *M. trifoliata* declined ~21%  
 204 (50-75% mean cover) by 2019, while cover of exotic *Mentha aquatica* L. increased ~385% (~25-50% mean

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 x for each result,  
 give evidence that  
 data met model  
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 that if rare sp. in  
 the plots weren't  
 counted, statement  
 about loss must to  
 be qualified.

209 cover). In summary, while the dominant species are maintained, their cover abundance within each assemblage  
 210 declined. Moreover, some exotic species have increased substantially in cover abundance in the Fescue and  
 211 Bogbean assemblages since the original 1979 survey. Increasing abundance of exotic species within each  
 212 assemblage is likely driving the greater similarity within assemblages (homogenization) and greater dissimilarity  
 213 between assemblages, as shown by cluster analysis (Figure 2).

214

215 *Table 1 Between 1979 and 2019, 8 fewer plots and 5 fewer species were observed, resulting in lower  $\alpha$ -diversity and greater  $\beta$ -diversity.*  
 216 *For each assemblage type, Bogbean is the only assemblage to proportionally gain plots between 1979 and 2019, while the Fescue and*  
 217 *Sedge assemblages lost plots. Plot loss did not appear to have an effect on diversity components, as tested by bootstrapping a minimum*  
 218 *of 18 plots per assemblage each year (Table 5)*

Assemblage	Plot-level components		Diversity components		
	No. quadrats	No. species	$\alpha$ diversity	$\alpha$ diversity sd	$\beta$ diversity
<b>Sedge</b>					
1979	34	36	8.74	2.45	3.89
1999	31	35	8.26	1.98	4.24
2019	28	34	7.89	2.69	4.31
<b>Fescue</b>					
1979	29	47	12.83	3.87	3.66
1999	33	41	9.69	3.96	4.23
2019	18	27	5.83	2.79	4.63
<b>Bogbean</b>					
1979	19	32	12.84	3.61	2.49
1999	18	36	11.50	2.92	3.13
2019	28	34	10.46	1.90	3.25
<b>Total</b>					
1979	82	48	9.96	3.41	4.82
1999	82	45	9.55	3.30	4.71
2019	74	43	8.36	3.03	5.14

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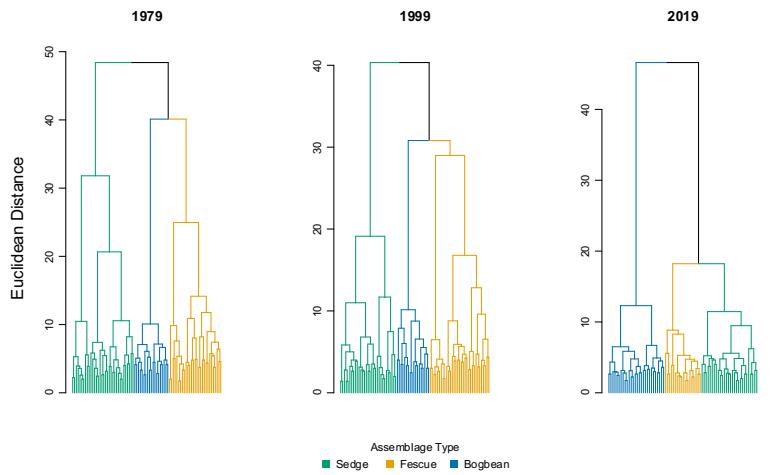
Abundance?

- rarefaction  
 or Shannon?  
 These would allow  
 some consideration  
 of how the community  
 is assembled.

223    Table 2. Species significantly driving cluster groups (Euclidean distance) include the same dominant species in each assemblage type  
 224    (Sedge by *Carex lyngbyei*, Fescue by *Festuca arundinaceae*, Bogbean by *Menyanthes trifoliata*). Indicator species significantly defining the  
 225    assemblage reported for  $p < 0.05$ .

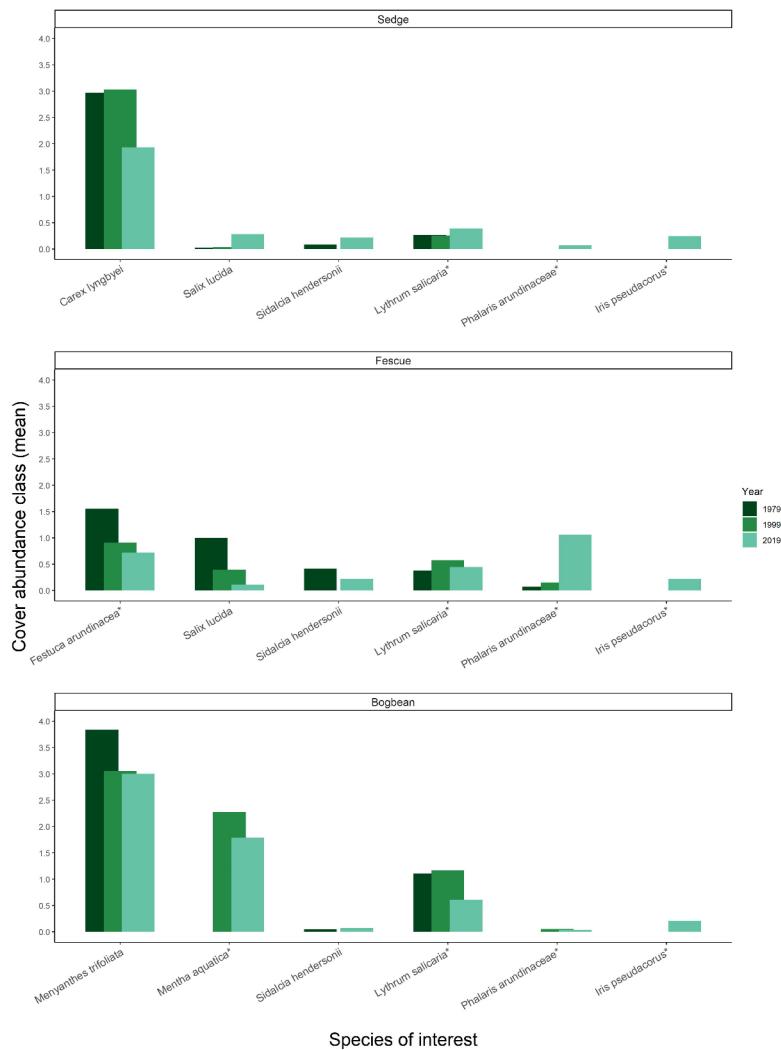
Cluster Group Name	1979		1999		2019	
	Species	p-value	Species	p-value	Species	p-value
"Sedge"	<i>Carex lyngbyei</i>	0.0001	<i>Carex lyngbyei</i>	0.0001	<i>Carex lyngbyei</i>	0.0001
	<i>Sagittaria latifolia</i>	0.0001	<i>Impatiens capensis</i>	0.0147		
	<i>Schoenoplectus tabernaemontani</i>	0.0004				
"Fescue"	<i>Festuca arundinacea</i>	0.0001	<i>Poa palustris</i>	0.0001	<i>Phalaris arundinacea</i>	0.0001
	<i>Salix lasiandra</i>	0.0001	<i>Festuca arundinacea</i>	0.0006	<i>Festuca arundinacea</i>	0.0001
	<i>Equisetum palustre</i>	0.0001	<i>Trifolium wormskjoldii</i>	0.0014	<i>Equisetum fluviatile</i>	0.0127
	<i>Lathyrus palustris</i>	0.0003	<i>Bidens cernua</i>	0.0044		
	<i>Sidalcea hendersonii</i>	0.0058				
	<i>Hordeum brachyantherum</i>	0.0157				
	<i>Deschampsia caespitosa</i>	0.0455				
"Bogbean"	<i>Menyanthes trifoliata</i>	0.0001	<i>Mentha aquatica</i>	0.0001	<i>Menyanthes trifoliata</i>	0.0001
	<i>Myosotis scorpioides</i>	0.0003	<i>Menyanthes trifoliata</i>	0.0001	<i>Mentha aquatica</i>	0.0001
	<i>Bidens cernua</i>	0.0012	Grass (unidentified)	0.0005	<i>Lysimachia thyrsiflora</i>	0.0001
	<i>Lythrum salicaria</i>	0.0012	<i>Lythrum salicaria</i>	0.0012	<i>Galium trifidum</i>	0.0006
	<i>Equisetum fluviatile</i>	0.0106	<i>Juncus articulatus</i>	0.0005	<i>Myosotis scorpioides</i>	0.0056
	<i>Lysimachia thyrsiflora</i>	0.0103	<i>Equisetum fluviatile</i>	0.0016	<i>Juncus articulatus</i>	0.0151
			<i>Myosotis scorpioides</i>	0.0046		
			<i>Eleocharis palustris</i>	0.0224		
			<i>Equisetum variegatum</i>	0.0447		
			<i>Deschampsia caespitosa</i>	0.0270		

I worry about circular reasoning here. Were you & previous researchers looking for clusters in the field? The description of Sampling methods makes me think Sampling was not random.



227

228 *Figure 2 Species cover abundance becomes more dissimilar in each assemblage over time, as shown by greater Euclidean distance*  
229 *between assemblage types. Note clusters of the sedge and fescue assemblages are more similar in 2019*



make fruits larger  
for the juncel.

230

Species of interest

231 *Figure 3 Assemblage-defining species within each assemblage have decreased in abundance over time, while several exotic species (denoted by \*) have increased in cover abundance since 1979. Cover classes are: [1] = < 25%, [2] = 25-50%, [3] = 51-75%, [4] = >75% above-ground vegetated cover.*

232  
233

## 234 Discussion

235 Despite conservation status and general resilience of the Fraser River marsh ecosystem we  
 236 found substantive changes in species composition over a 40 year time-frame, potentially indicating  
 237 broader-scale processes affected by regional pressures. The three species most significantly  
 238 characterizing the three plant assemblages, Sedge, Fescue and Bogbean, have remained the same over  
 239 the past 40 years, supporting our expectation that these characteristic species should not change in the  
 240 absence of significant disturbance. We observed a decline of native species richness accompanied by an  
 241 increased richness and abundance of exotic species, which may indicate a loss of community stability.  
 242 This potential instability may further be evidenced by the homogenization of cover abundance within  
 243 assemblages and overall loss of indicator species for the Sedge and Fescue assemblages. Our results  
 244 present a compelling case example of broader global trends of native species biodiversity loss, and  
 245 should be of concern to estuary managers whose objectives are to conserve wildlife habitat and  
 246 intertidal shoreline stability.

247 The changing identity of species or functional traits in an assemblage may offer clues to shifting  
 248 abiotic conditions within or between assemblages (Waller et al., 2020). One functional group to note  
 249 were the woody species, as their traits convey different structural habitat qualities than herbaceous  
 250 species. Willow (*Salix lucida* Muhl.) was most prevalent in the Fescue assemblage in 1979, but was most  
 251 abundant in the Sedge assemblage in 2019. This could suggest long-term shifts in edaphic factors and/or  
 252 the competitive encroachment of exotic reed canary grass (*Phalaris arundinaceae*), making the Fescue  
 253 assemblage less hospitable to willow recruitment. Alternatively, this could indicate that environmental  
 254 conditions are becoming more similar between the two assemblages, as evidenced by the clustering of  
 255 the Fescue and Sedge groups on the same branch in the 2019 dendrogram (Figure 2). The indicator  
 256 species analysis for the Sedge assemblage in 1979 included plants tolerant of highly saturated soils  
 257 (*Sagittaria latifolia*, *Schoenoplectus tabernaemontani*), but in 1999 the assemblage indicators included  
 258 species less tolerant of aquatic or constantly saturated soils (*Impatiens capensis*) ([Error! Reference  
source not found.](#)).

260 In contrast, the turnover of indicator species may simply represent high interannual variation in  
 261 species compositional abundance, despite being a perennial-dominated community. For example, the  
 262 Bogbean assemblage, was indicated largely by unique forbs in 1979 and 2019, and an even mix of  
 263 unique forbs and graminoids in 1999 ([Error! Reference source not found.](#)). In the Bogbean assemblage  
 264 it is harder to ascribe inference, such as potential woody riparian succession, to the replacement of forb  
 265 indicator species as in the Sedge and Fescue assemblages. The indicator graminoid species found only in  
 266 1999 in the Bogbean assemblage (excluding an unknown grass identified only to family) are all native  
 267 wetland species commonly found in high marshes. Rather than indicating altered abiotic conditions,  
 268 their inclusion as indicator species may represent population dynamics of short-lived perennials such as  
 269 dispersal and recruitment. Thus, we propose two potential alternative hypotheses for the observed  
 270 changes in floristic composition observed in the different assemblages: greater compositional  
 271 abundance of woody species or species tolerant of drier conditions could be indicative of channel  
 272 morphology processes limiting bank topography suitable for aquatic emergent plants, or sedimentation  
 273 feedback processes increasing elevation of the marsh platform relative to tidal inundation. Alternatively,  
 274 population dynamics may be operating independently of abiotic conditions, or have different outcomes  
 275 depending on edaphic conditions in each assemblage. Testing how life histories (e.g., species longevity)  
 276 offer competitive advantage in the context of changing abiotic conditions would be a valuable long-term

explanations

For natural species  
turnover

I think this needs a  
reference to a gro<sup>2</sup>

**Deleted:** secondary

theory paper AND  
an empirical  
example

→ the evidence  
for species loss is  
not strong.

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inference is the  
conclusion from  
the stats. Do you  
mean attribution?

Are you sure  
this hasn't  
been done?

279 addition to general interactions of competition and edaphic factors. These interactions would present a  
280 valuable experimental test of competitive advantage or how edaphic conditions drive the dominance of  
281 native vs. exotic species in tidal wetlands.

282 Greater homogeneity of cover abundance within assemblages, and greater distinction in  
283 compositional abundance between assemblages, may result from overall loss of native floristic richness.  
284 Across all assemblages in Ladner Marsh 1979–2019, plots had on average one to two fewer native  
285 species, while  $\beta$ -diversity increased. This would indicate that rare (infrequently found) species are  
286 becoming more locally rare, which contributes to the loss of heterogeneous cover abundance and  
287 increased  $\beta$ -diversity observed at the plot scale. With the exception of woody species gained, the life  
288 histories and ecosystem functional traits of the species gained are similar to those lost from Ladner  
289 Marsh. More concerning is the net loss of six perennial graminoid and forb species over the study  
290 period, as this represents a loss of functional redundancy from the ecosystem. Biodiversity loss may  
291 reduce the dense root networks to trap sediment in the marsh platform and seasonal pollinator value of  
292 forbs, although these contributions by the species lost in Ladner Marsh have not been quantified.  
293 Regardless of whether the loss is due to turnover or shifting abiotic conditions, trends of lost species  
294 richness may indicate greater susceptibility to invasion (Kuiters, et al., 2009), and thus a loss of  
295 resistance to exotic species encroachment over time. This can be evidenced by the decreasing ratio of  
296 native to exotic cover across Ladner Marsh 1979–2019 (Figure 5), although few species (native or exotic)  
297 represent the majority of cover within the assemblage (Table 7). Exotic species of significant  
298 management concern (e.g., *P. arundinaceae*, *I. pseudacorus*) were  $\leq 25\%$  mean plot cover in 2019,  
299 however these species are notorious for spreading to the point of near-exclusion of other species  
300 (especially natives) (Apfelbaum & Sams, 1987; Sinks et al., 2021).

### 301 Mechanisms, Synthesis & Recommendations

302 Exotic species encroachment and native species loss may lead to instability in native populations  
303 through fragmented or lost propagule dispersal networks, resulting in ecosystem instability through  
304 altered trophic cascades and implications for endangered species. Disentangling explicit impacts of  
305 sedimentation, propagule dispersal, or propagule recruitment processes would be no easy task in a tidal  
306 ecosystem, however experimentally testing optimal recruitment niches of species-specific propagules  
307 (e.g., Lane, 2022) could prove valuable for understanding best practices to maintain at-risk populations  
308 or test community function. Optimal abiotic conditions for the recruitment and spatial occupancy of  
309 native or exotic species may largely be driven by soil characteristics and related sedimentation  
310 processes. Sedimentary changes such as sediment starvation or subsidence would result in more  
311 saturated areas, which would likely drive the increased prevalence of saturated conditions favored by  
312 the Bogbean assemblage (Mendelsohn & Kuhn, 2003). Alternatively, positive feedbacks between  
313 vegetation and sedimentation could support areas of marsh accretion (Nyman et al., 2006), which may  
314 also be more likely to receive exotic propagules within the distributed sediment. Propagule pools would  
315 depend on local and regional proximity. If similar habitats within tidal estuarine ecosystems are lost to  
316 the point where distance between patches exceeds propagule dispersal distance (Shi, et al., 2020), then  
317 species colonization within the ecosystem is rare or lost. Alternatively, if exotic species are more  
318 prevalent throughout the regional dispersal network, then there is a greater chance of exotic species  
319 introduction within a local marsh community. Thus, abiotic shifts may be altering the seed recruitment  
320 niches which may restrict recruitment of native species diversity, while dispersal networks may be  
321 delivering disproportionately more seed of exotic species. This reflects a general trend of exotic species'

I'm curious what has actually  
happened at this site.  
accretion? Climate change?  
etc? otherwise this is  
V. speculative.

again, w/ some  
assurance that  
missed sp. were  
estimated & sampling  
effort was controlled  
for, this claim  
is not supported.

→ how do you  
know they were  
not at the site  
but just not  
in the plot?

need refs for this o/p next  
it's not supposed. There  
are many examples where  
this statement is not true.

322 competitive advantage in disturbed systems, and represents ongoing press disturbance by  
323 anthropogenic impacts with cumulative ecosystem effects.

324 A common (mis)assumption is that 'undisturbed' areas represent ecologically appropriate  
325 reference states (e.g., Stoddard, et al., 2006, and citations therein). Our findings support that  
326 contemporary "reference" sites are not sufficient benchmarks for restoration success (Shackelford, et  
327 al., 2021). The biodiversity loss described here presents real concerns for this conservation area, and  
328 provides another case example of negative biodiversity trends in habitat considered relatively pristine.  
329 Active management informed by experimental testing of hydrogeomorphologic drivers, dispersal  
330 networks, and recruitment strategies will be needed to maintain ecologically desired species  
331 composition in the face of climate change. If we are to prioritize conservation of functional coastal  
332 wetlands that include a significant representation of native species, we must seek new ways to actively  
333 manage habitats such as the Ladner Marsh. Through control of invasive species and experimental  
334 management practices to employ sediment application and/or native species planting, practitioners may  
335 enhance ecosystem processes within remnant coastal wetland habitats. This active management  
336 process also presents a timely and necessary opportunity in the Pacific Northwest of North America to  
337 engage with First Nations to revive traditional management practices in tidal wetlands, such as select  
338 mechanical disturbance (Turner, 2014): working with traditional knowledge holders in these ecosystems  
339 may yield deeper understanding of plant community function and habitat stability, which would  
340 enhance ecosystem resilience and potentially lead to positive effects on regionally important salmonid  
341 and shorebird populations while contributing to reconciliation between Indigenous and colonial  
342 cultures.

343

or, is this a  
? Control site that  
could be used for  
comparison to an intact,  
natural marsh if it's  
ever frozen in  
that state

344 **Statements & Declarations**

345

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351 **Competing interests**

352 The authors have no relevant financial or non-financial interests to disclose.

353 **Author contributions**

354 Study conception, design, 2019 data collection, and analysis were exclusively undertaken by Stefanie L.  
355 Lane. Original (1979) study concept comparing plant assemblages, data collection, and analysis were  
356 performed or overseen by Gary Bradfield. Madlen Denoth contributed data collected in 1999. Nancy  
357 Shackelford assisted with theoretical framework and manuscript revision. Manuscript was drafted by  
358 Stefanie L. Lane; Nancy Shackelford and Tara G. Martin commented on previous versions of the  
359 manuscript. All authors read and approved the final manuscript.

360 **Data availability**

361 Data for all years of observation are available on Dryad (DOI). Code is available on GitHub (REPO), or via  
362 Dryad (DOI)

363

364

**Commented [SLL11]:** Pending permission from Gary and/or Madlen

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468 Supplemental  
469470 *Table 3.* A total of 25 plots sampled in 1979 and 1999 were not sampled in 2019, mostly due to issues of accessibility. Transect  
471 names and plot ID of plots omitted follow Fig. 3 in Bradfield & Porter (1982).

Transect	1979/1999 Plot No.	Reason omitted in 2019
Q	1-7	Transect in dense riparian thicket overgrown with Himalayan blackberry
R	8	Plot on lower bench (> 1 m lower than marsh platform), vegetation no longer exists
R	17-19	Plots in 1979 & 1999 sampled across a channel. Ended transect in 2019 at channel edge.
S	33-36	Transect length in 2019 was shorter than in 1979/1999. Suspect combination of erosion and offset transect relocation altered sampling distance.
T	45	Transect length in 2019 was shorter than in 1979/1999. Suspect combination of erosion and offset transect relocation altered sampling distance.
U	51-52	Transect length in 2019 was shorter than in 1979/1999. Suspect combination of erosion and offset transect relocation altered sampling distance.
V	53	Plot 53 only plot across a channel. Increased channel width and likely erosion made crossing this channel dangerous; omitted plot in 2019.
V	54, 70-71	Transect length in 2019 was shorter than in 1979/1999. Suspect combination of erosion and offset transect relocation altered sampling distance.
W	89-92	Transect length in 2019 was shorter than in 1979/1999. Suspect combination of erosion and offset transect relocation altered sampling distance.
X	93	Transect length in 2019 was shorter than in 1979/1999. Suspect combination of erosion and offset transect relocation altered sampling distance.

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475    Table 4. Species indicator analysis of cluster groups using Bray-Curtis distance identifies the same dominant species in each assemblage type (Sedge, Fescue, Bogbean), however Bray-Curtis  
 476    distance identifies different associated indicator species than those identified by Euclidean distance ([Error! Reference source not found.](#)).

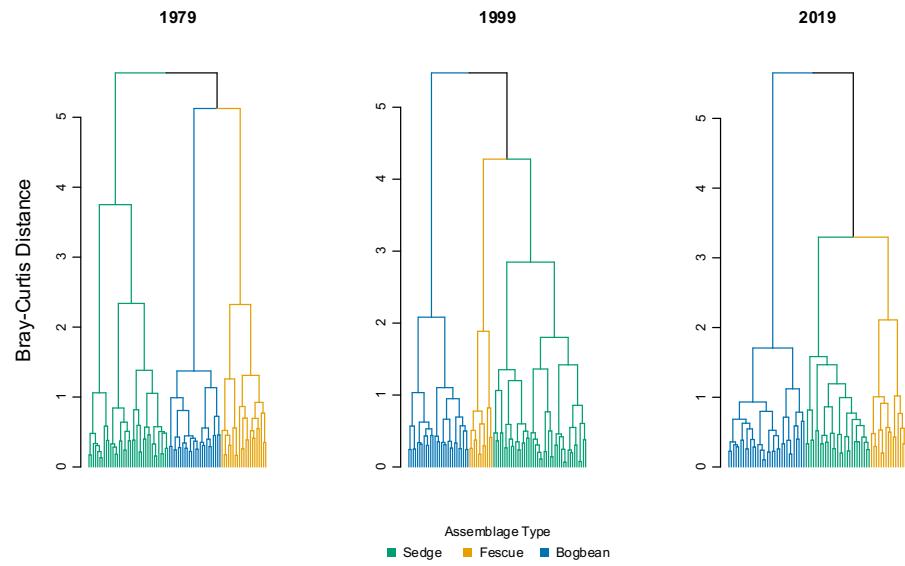
477

Cluster Group Name	Species	1979		1999		2019			
		IndVal stat	p-value	IndVal stat	p-value	IndVal stat	p-value		
"Sedge"	<i>Carex lyngbyei</i>	0.678	0.001	<i>Carex lyngbyei</i>	0.804	0.001	<i>Carex lyngbyei</i>	0.714	0.001
	<i>Sagittaria latifolia</i>	0.559	0.001	<i>Agrostis stolonifera</i>	0.434	0.003	<i>Mentha arvensis</i>	0.322	0.033
	<i>Schoenoplectus tabernaemontani</i>	0.391	0.001						
"Fescue"	<i>Festuca arundinacea</i>	0.753	0.001	<i>Festuca arundinacea</i>	0.765	0.001	<i>Phalaris arundinaceae</i>	0.584	0.001
	<i>Salix lucida</i>	0.586	0.001	<i>Phalaris arundinaceae</i>	0.334	0.019	<i>Festuca arundinacea</i>	0.416	0.001
	<i>Lathyrus palustris</i>	0.543	0.001						
	<i>Equisetum palustre</i>	0.475	0.002						
	<i>Impatiens capensis</i>	0.391	0.002						
	<i>Sidalcea hendersonii</i>	0.387	0.001						
	<i>Platanthera dilatata</i>	0.308	0.020						
"Bogbean"	<i>Menyanthes trifoliata</i>	0.807	0.001	<i>Menyanthes trifoliata</i>	0.782	0.001	<i>Mentha aquatica</i>	0.752	0.001
	<i>Myosotis scorpioides</i>	0.577	0.001	<i>Leersia oryzoides</i>	0.495	0.001	<i>Menyanthes trifoliata</i>	0.709	0.001
	<i>Juncus articulatus</i>	0.523	0.001	<i>Mentha aquatica</i>	0.492	0.001	<i>Lysimachia thyrsiflora</i>	0.547	0.001
	<i>Lythrum salicaria</i>	0.400	0.002	<i>Bidens cernua</i>	0.489	0.003	<i>Salix lucida</i>	0.465	0.001
	<i>Lysimachia thyrsiflora</i>	0.400	0.002	<i>Lysimachia thyrsiflora</i>	0.478	0.001	<i>Eleocharis palustris</i>	0.460	0.001
	<i>Trifolium wormskoldii</i>	0.381	0.003	<i>Juncus articulatus</i>	0.438	0.001	<i>Juncus articulatus</i>	0.373	0.004
	<i>Lilaeopsis occidentalis</i>	0.360	0.004	<i>Juncus oxymeris</i>	0.356	0.015	<i>Galium trifidum</i>	0.348	0.008
	<i>Mentha aquatica</i>	0.313	0.010	<i>Myosotis scorpioides</i>	0.356	0.019	<i>Bidens cernua</i>	0.323	0.012
				<i>Poaceae (unidentified sp.)</i>	0.356	0.013			
				<i>Deschampsia caespitosa</i>	0.354	0.014			
				<i>Sagittaria latifolia</i>	0.301	0.046			

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482 *Figure 4. Cluster analysis using Bray-Curtis distance measure shows similar trends of increasing dissimilarity over time as when using*  
483 *Euclidean distance (Figure 2).*

484

485     Table 5. Bootstrapping 18 randomly selected plots 10 times shows consistent overall trend in loss of species and alpha diversity over time,  
 486     and overall increase in beta diversity between 1979 and 2019 in all assemblages and across the entire Ladner Marsh plant community.  
 487     Therefore, loss of plots due to sampling re-location or how number of plots clustered into assemblages as reported in Table 2 is not  
 488     expected to affect loss of species or plot-based diversity metrics.

	Plot-level components			Diversity components		
	Assemblage	No. quadrats	No. species	$\alpha$ diversity	$\alpha$ diversity sd	$\beta$ diversity
<b>Sedge</b>						
	1979	18	32.3	10.67	2.34	3.03
	1999	18	31.6	8.31	1.98	3.81
	2019	18	30.8	8.18	2.51	3.77
<b>Fescue</b>						
	1979	18	43.3	12.96	3.91	3.35
	1999	18	36.4	9.72	3.92	3.76
	2019	18	27	5.83	2.79	4.63
<b>Bogbean</b>						
	1979	18	31.7	12.83	3.63	2.47
	1999	18	36	11.50	2.92	3.13
	2019	18	31.4	10.52	1.90	2.99
<b>Total</b>						
	1979	54	47.6	12.15	3.49	3.92
	1999	54	42.1	10.02	3.35	4.22
	2019	54	41.7	8.18	3.08	5.10

506     Table 6. Total turnover and rates of species disappearance (loss) was always greater between 1999 and 2019 than between 1979 and  
 507     1999. However, fewer species were gained in the Bogbean assemblage 1999-2019 than 1979-1999.

Assemblage	Year	Total turnover	Species Appearance	Species Disappearance
Bogbean	1979-1999	0.56	0.35	0.22
	1999-2019	0.60	0.28	0.32
Fescue	1979-1999	0.46	0.20	0.27
	1999-2019	0.64	0.18	0.46
Sedge	1979-1999	0.46	0.24	0.22
	1999-2019	0.56	0.27	0.29

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513 *Table 7. Percent change in mean abundance (cover class) between from 1979 to 2019 for non-native and native species observed in each*  
 514 *assemblage. New species appearances from 1979 to 2019 indicated by (+); species only appearing in 1999 indicated by 'NA'. Native status*  
 515 *is listed as 'unknown' if plant was not identified to species level.*

Assemblage	Status	Species	1979	1999	2019	Percent Change (1979-2019)
Bogbean	Non-native	<i>Mentha arvensis</i>	0.47	0.00	0.07	-84.9
		<i>Myosotis scorpioides</i>	0.68	0.22	0.21	-68.7
		<i>Agrostis stolonifera</i>	3.21	1.50	1.29	-60.0
		<i>Lythrum salicaria</i>	1.11	1.17	0.61	-45.1
		<i>Rumex conglomeratus</i>	0.05	0.00	0.04	-32.1
		<i>Mentha aquatica</i>	0.37	2.28	1.79	384.7
		<i>Iris pseudocorus</i>	0.00	0.33	0.21	+
		<i>Lycopus europaeus</i>	0.00	0.00	0.04	+
		<i>Phalaris arundinacea</i>	0.00	0.06	0.04	+
		<i>Festuca arundinacea</i>	0.00	0.17	0.00	NA
	Native	<i>Alisma plantago aquatica</i>	0.16	0.11	0.00	-100.0
		<i>Alopecurus geniculatus</i>	0.05	0.00	0.00	-100.0
		<i>Deschampsia caespitosa</i>	0.26	0.22	0.00	-100.0
		<i>Equisetum fluviatile</i>	1.37	1.17	0.00	-100.0
		<i>Leersia oryzoides</i>	0.26	0.33	0.00	-100.0
		<i>Lilaeopsis occidentalis</i>	0.21	0.00	0.00	-100.0
		<i>Oenanthe sarmentosa</i>	0.63	0.11	0.00	-100.0
		<i>Poa trivialis</i>	0.11	0.00	0.00	-100.0
		<i>Sium suave</i>	0.63	0.17	0.00	-100.0
		<i>Caltha palustris</i>	0.95	0.22	0.07	-92.5
		<i>Bidens cernua</i>	0.84	0.17	0.14	-83.0
		<i>Trifolium wormskoldii</i>	0.95	0.11	0.18	-81.2
		<i>Schoenoplectus tabernaemontani</i>	0.16	0.00	0.07	-54.8
		<i>Eleocharis palustris</i>	0.63	0.78	0.39	-37.8
		<i>Sympyotrichum subspicatum</i>	0.47	0.33	0.32	-32.1
		<i>Juncus oxymeris</i>	0.05	0.11	0.04	-32.1
		<i>Platanthera dilatata</i>	0.05	0.06	0.04	-32.1
		<i>Menyanthes trifoliata</i>	3.84	3.06	3.00	-21.9
		<i>Lysimachia thyrsiflora</i>	0.53	0.22	0.57	8.6
		<i>Juncus articulatus</i>	0.26	0.39	0.29	8.6
		<i>Sidalcea hendersonii</i>	0.05	0.00	0.07	35.7
		<i>Carex lyngbyei</i>	0.47	0.33	1.00	111.1
		<i>Rumex occidentalis</i>	0.05	0.11	0.14	171.4
		<i>Potentilla anserina-pacifica</i>	0.26	1.00	1.07	307.1
		<i>Equisetum arvense</i>	0.00	0.00	0.64	+
		<i>Galium trifidum</i>	0.00	0.00	0.39	+
		<i>Hypericum scouleri</i>	0.00	0.00	0.04	+

		<i>Impatiens capensis</i>	0.00	0.44	0.32	+
		<i>Juncus acuminatus</i>	0.00	0.00	0.04	+
		<i>Lathyrus palustris</i>	0.00	0.11	0.50	+
		<i>Lysichiton americanum</i>	0.00	0.00	0.07	+
		<i>Salix lasiandra</i>	0.00	0.61	0.50	+
		<i>Salix scouleriana</i>	0.00	0.00	0.04	+
		<i>Typha latifolia</i>	0.00	0.28	0.25	+
		<i>Equisetum palustre</i>	0.00	0.11	0.00	NA
		<i>Equisetum variegatum</i>	0.00	0.11	0.00	NA
		<i>Galium sp.</i>	0.00	0.06	0.00	NA
		<i>Poa palustris</i>	0.00	0.50	0.00	NA
		<i>Poaceae sp.</i>	0.00	0.28	0.00	NA
		<i>Sagittaria latifolia</i>	0.00	0.17	0.00	NA

Assemblage	Status	Species	1979	1999	2019	Percent Change (1979-2019)
Fescue	Non-native	<i>Festuca sp.</i>	0.03	0.00	0.00	-100.0
		<i>Mentha aquatica</i>	0.31	0.09	0.00	-100.0
		<i>Myosotis scorpioides</i>	0.31	0.03	0.00	-100.0
		<i>Mentha arvensis</i>	0.17	0.24	0.06	-67.8
		<i>Festuca arundinacea</i>	1.55	0.91	0.72	-53.5
		<i>Lythrum salicaria</i>	0.38	0.58	0.44	17.2
		<i>Agrostis stolonifera</i>	0.34	0.82	0.61	77.2
		<i>Phalaris arundinacea</i>	0.07	0.15	1.06	1430.6
		<i>Cirsium arvense</i>	0.00	0.03	0.06	+
		<i>Iris pseudocorus</i>	0.00	0.15	0.22	+
	Native	<i>Lycopus europaeus</i>	0.00	0.00	0.06	+
		<i>Alisma plantago aquatica</i>	0.10	0.18	0.00	-100.0
		<i>Alopecurus geniculatus</i>	0.03	0.00	0.00	-100.0
		<i>Bidens cernua</i>	0.21	0.52	0.00	-100.0
		<i>Deschampsia caespitosa</i>	0.62	0.09	0.00	-100.0
		<i>Dulichium arundinaceum</i>	0.07	0.00	0.00	-100.0
		<i>Eleocharis palustris</i>	0.97	0.33	0.00	-100.0
		<i>Equisetum palustre</i>	0.76	0.09	0.00	-100.0
		<i>Galium trifidum</i>	0.03	0.00	0.00	-100.0
		<i>Hypericum formosum</i>	0.10	0.00	0.00	-100.0
		<i>Juncus articulatus</i>	0.52	0.06	0.00	-100.0
		<i>Leersia oryzoides</i>	0.14	0.24	0.00	-100.0
		<i>Lilaeopsis occidentalis</i>	0.17	0.00	0.00	-100.0
		<i>Mimulus guttatus</i>	0.03	0.00	0.00	-100.0
		<i>Oenanthe sarmentosa</i>	0.17	0.30	0.00	-100.0
		<i>Platanthera dilatata</i>	0.21	0.03	0.00	-100.0
		<i>Poa palustris</i>	0.55	1.73	0.00	-100.0

<i>Poa trivialis</i>	0.31	0.00	0.00	-100.0
<i>Polygonum hydropiper</i>	0.03	0.00	0.00	-100.0
<i>Sagittaria latifolia</i>	0.03	0.15	0.00	-100.0
<i>Salix sp.</i>	0.03	0.00	0.00	-100.0
<i>Sium suave</i>	0.14	0.15	0.00	-100.0
<i>Sympyotrichum subspicatum</i>	0.59	0.24	0.00	-100.0
<i>Trifolium wormskioldii</i>	0.69	0.55	0.00	-100.0
<i>Menyanthes trifoliata</i>	1.86	1.33	0.06	-97.0
<i>Caltha palustris</i>	0.66	0.39	0.06	-91.5
<i>Salix lasiandra</i>	1.00	0.39	0.11	-88.9
<i>Carex lyngbyei</i>	0.76	1.42	0.11	-85.4
<i>Potentilla anserina-pacifica</i>	0.48	0.64	0.22	-54.0
<i>Sidalcea hendersonii</i>	0.41	0.18	0.22	-46.3
<i>Lysimachia thyrsiflora</i>	0.10	0.33	0.06	-46.3
<i>Typha latifolia</i>	0.69	0.36	0.44	-35.6
<i>Hordeum brachyantherum</i>	0.17	0.00	0.11	-35.6
<i>Equisetum fluviatile</i>	0.62	0.36	0.44	-28.4
<i>Schoenoplectus tabernaemontani</i>	0.07	0.15	0.06	-19.4
<i>Lathyrus palustris</i>	0.55	0.18	0.56	0.7
<i>Rumex occidentalis</i>	0.07	0.15	0.11	61.1
<i>Impatiens capensis</i>	0.28	0.42	0.61	121.5
<i>Equisetum arvense</i>	0.00	0.00	0.39	+
<i>Juncus effusus</i>	0.00	0.00	0.06	+
<i>Lysichiton americanum</i>	0.00	0.00	0.11	+
<i>Myrica gale</i>	0.00	0.00	0.22	+
<i>Salix scouleriana</i>	0.00	0.00	0.17	+
<i>Asteraceae sp.</i>	0.00	0.03	0.00	NA
<i>Carex sp.</i>	0.00	0.06	0.00	NA
<i>Galium sp.</i>	0.00	0.03	0.00	NA
<i>Juncus oxymeris</i>	0.00	0.09	0.00	NA
<i>Salix sitchensis</i>	0.00	0.03	0.00	NA

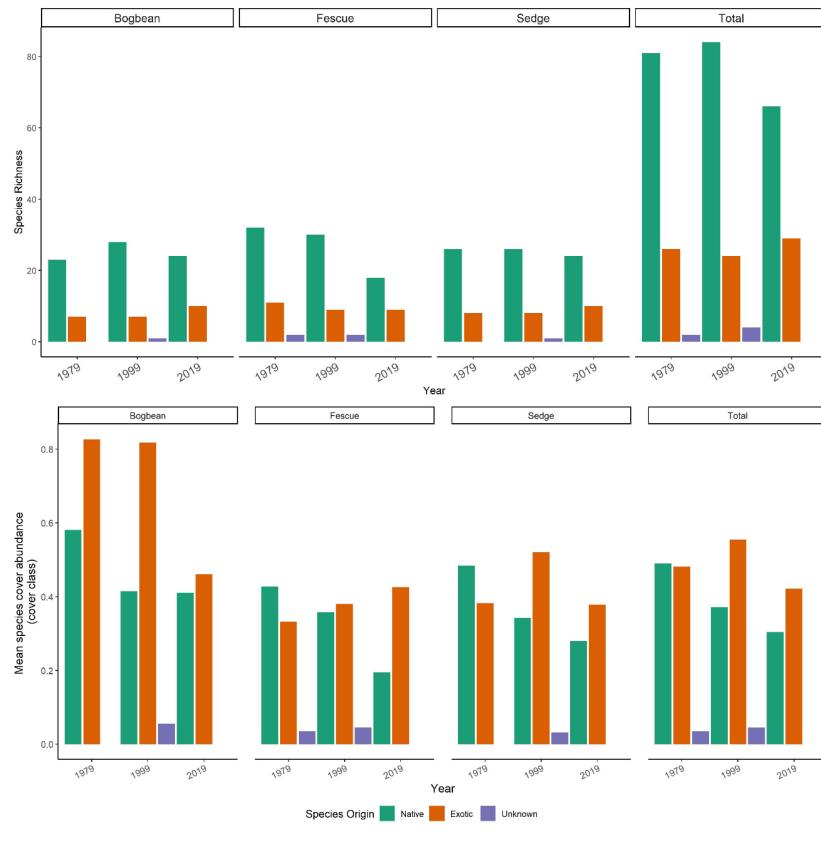
Assemblage	Status	Species	1979	1999	2019	Percent Change (1979-2019)
Sedge	Unknown	<i>Galium sp.</i>	0.00	0.03	0.00	NA
	Non-native	<i>Myosotis scorpioides</i>	0.03	0.00	0.00	-100.0
		<i>Mentha arvensis</i>	0.29	0.16	0.04	-87.9
		<i>Agrostis stolonifera</i>	1.85	2.32	1.25	-32.5
		<i>Lythrum salicaria</i>	0.26	0.26	0.39	48.4
		<i>Festuca arundinacea</i>	0.09	0.10	0.18	102.4
		<i>Iris pseudocorus</i>	0.00	0.13	0.25	+
		<i>Lycopus europaeus</i>	0.00	0.00	0.11	+
		<i>Mentha aquatica</i>	0.00	0.16	0.54	+

	<i>Phalaris arundinacea</i>	0.00	0.00	0.07	+
	<i>Cirsium arvense</i>	0.00	0.03	0.00	NA
Native	<i>Alisma plantago aquatica</i>	0.35	0.06	0.00	-100.0
	<i>Deschampsia caespitosa</i>	0.21	0.00	0.00	-100.0
	<i>Leersia oryzoides</i>	0.18	0.19	0.00	-100.0
	<i>Lilaeopsis occidentalis</i>	0.06	0.10	0.00	-100.0
	<i>Mimulus guttatus</i>	0.09	0.00	0.00	-100.0
	<i>Oenanthe sarmentosa</i>	0.71	0.39	0.00	-100.0
	<i>Platanthera dilatata</i>	0.09	0.03	0.00	-100.0
	<i>Poa palustris</i>	1.00	0.23	0.00	-100.0
	<i>Puccinella pauciflora</i>	0.03	0.00	0.00	-100.0
	<i>Sium suave</i>	0.59	0.19	0.00	-100.0
	<i>Caltha palustris</i>	1.09	0.48	0.04	-96.7
	<i>Equisetum fluviatile</i>	0.88	0.58	0.04	-96.0
	<i>Schoenoplectus tabernaemontani</i>	0.71	0.10	0.11	-84.8
	<i>Trifolium wormskoldii</i>	0.41	0.13	0.07	-82.7
	<i>Sagittaria latifolia</i>	0.41	0.10	0.11	-74.0
	<i>Bidens cernua</i>	0.47	0.13	0.21	-54.5
	<i>Eleocharis palustris</i>	0.79	0.35	0.39	-50.5
	<i>Menyanthes trifoliata</i>	0.32	0.68	0.18	-44.8
	<i>Carex lyngbyei</i>	2.97	3.03	1.93	-35.1
	<i>Typha latifolia</i>	0.59	0.35	0.43	-27.1
	<i>Sympyotrichum subspicatum</i>	0.29	0.13	0.25	-15.0
	<i>Rumex occidentalis</i>	0.12	0.16	0.11	-8.9
	<i>Lysimachia thyrsiflora</i>	0.09	0.00	0.11	21.4
	<i>Sidalcea hendersonii</i>	0.09	0.10	0.21	142.9
	<i>Potentilla anserina-pacifica</i>	0.29	0.74	0.79	167.1
	<i>Rumex conglomeratus</i>	0.03	0.00	0.11	264.3
	<i>Lathyrus palustris</i>	0.09	0.26	0.46	426.2
	<i>Impatiens capensis</i>	0.15	1.06	0.86	482.9
	<i>Salix lasiandra</i>	0.03	0.03	0.29	871.4
	<i>Equisetum arvense</i>	0.00	0.00	0.68	+
	<i>Galium palustre</i>	0.00	0.00	0.04	+
	<i>Galium trifidum</i>	0.00	0.00	0.07	+
	<i>Hypericum scouleri</i>	0.00	0.00	0.07	+
	<i>Juncus articulatus</i>	0.00	0.00	0.04	+
	<i>Juncus oxymeris</i>	0.00	0.00	0.04	+
	<i>Scirpus microcarpus</i>	0.00	0.00	0.07	+
	<i>Equisetum palustre</i>	0.00	0.19	0.00	NA
	<i>Lysichiton americanum</i>	0.00	0.03	0.00	NA
	<i>Salix sitchensis</i>	0.00	0.06	0.00	NA

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Figure 5. Top panel: Loss of native species richness over time across all assemblages is largely driven by loss of native species from the Fescue Assemblage. However, native species richness does not change substantially in the other two assemblages. Bottom panel: Native species cover is decreasing on average across all assemblages. Exotic species cover largely remains unchanged, although the ratio of native to exotic cover in Bogbean assemblage becomes more even by 2019. 'Unknown' species origin represents species identified only to genus, and assessment of native status cannot be made.

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530 Table 8. All species recorded in 1979, 1999, and 2019, their synonymous nomenclature, and endemic status according to the United States  
 531 Department of Agriculture Natural Resources Conservation Service PLANTS Database.

<b>Species found 1979-2019</b>	<b>Synonym recorded in 1979, 1999</b>	<b>Endemism Status</b>
<i>Agrostis stolonifera</i>	<i>Agrostis alba</i>	Exotic
<i>Alisma plantago-aquatica</i>		Exotic
<i>Alopecurus geniculatus</i>		Exotic
<i>Bidens cernua</i>		Native
<i>Caltha palustris</i>		Native
<i>Carex lyngbyei</i>		Native
<i>Carex sp1</i>		NA
<i>Carex sp2</i>		NA
<i>Cirsium arvense</i>		Exotic
Composite (unidentified)		NA
<i>Deschampsia caespitosa</i>		Native
<i>Dulichium arundinaceum</i>		Native
<i>Eleocharis palustris</i>		Native
<i>Equisetum arvense</i>		Native
<i>Equisetum fluviatile</i>		Native
<i>Equisetum variegatum</i>		Native
<i>Festuca arundinaceaee</i>		Exotic
<i>Festuca</i> sp		NA
<i>Galium palustre</i>		Native
<i>Galium</i> sp		NA
<i>Galium trifidum</i>	<i>Galium cymosum</i>	Native
Grass (unidentified)		NA
<i>Hordeum brachyantherum</i>		Native
<i>Hypericum scouleri</i>	<i>Hypericum formosum</i>	Native
<i>Impatiens capensis</i>		Exotic
<i>Iris pseudacorus</i>		Exotic
<i>Juncus acuminatus</i>		Native
<i>Juncus articulatus</i>		Native
<i>Juncus effusus</i>		Native
<i>Juncus oxymeris</i>		Native
<i>Lathyrus palustris</i>		Native
<i>Leersia oryzoides</i>		Native
<i>Lilaea scilloides</i>		Native
<i>Lilaeopsis occidentalis</i>		Native
<i>Lycopus europaeus</i>		Exotic
<i>Lysichiton americanus</i>		Native
<i>Lysimachia thrysiflora</i>		Native
<i>Lythrum salicaria</i>		Exotic
<i>Mentha aquatica</i>	<i>Mentha citrata</i>	Exotic
<i>Mentha arvensis</i>		Exotic
<i>Menyanthes trifoliata</i>		Native

<i>Mimulus guttatus</i>	Native	
<i>Myosotis scorpioides</i>	Exotic	
<i>Myrica gale</i>	Native	
<i>Oenanthe sarmentosa</i>	Native	
<i>Phalaris arundinacea</i>	Exotic	
<i>Platanthera dilatata var dilatata</i>	Native	
<i>Poa palustris</i>	Native	
<i>Poa trivialis</i>	Exotic	
<i>Polygonum hydropiper</i>	Exotic	
<i>Potentilla pacifica</i>	Native	
<i>Puccinellia pauciflora</i>	Native	
<i>Rumex conglomeratus</i>	Exotic	
<i>Rumex occidentalis</i>	Native	
<i>Sagittaria latifolia</i>	Native	
<i>Salix lasiandra</i>	Native	
<i>Salix scouleriana</i>	Native	
<i>Salix sitchensis</i>	Native	
<i>Salix</i> sp	NA	
<i>Schoenoplectus tabernaemontani</i>	<i>Scirpus validus</i>	Native
<i>Scirpus microcarpus</i>	Native	
<i>Sidalcea hendersonii</i>	Native	
<i>Sium suave</i>	Native	
<i>Sonchus arvensis</i>	Exotic	
<i>Sympotrichum subspicatum</i>	<i>Aster eatonii</i>	Native
<i>Trifolium wormskiioldii</i>	<i>Trifolium wormskjoldii</i>	Native
<i>Typha latifolia</i>	Native	
<i>Zannichellia palustris</i>	Native	

## Potential journals

### Wetlands

2020 SJR IF 2.369 (Q2, Ecology)

Original research: Articles reporting original research about wetlands, natural or constructed, including, but not limited to mechanisms underlying ecosystem processes, the values of wetlands to society, their management, **quality assessment** and restoration.

### Marine & Freshwater Research

2020 SJR IF 2.034 (Q2, Aquatic Science)

Marine and Freshwater Research welcomes the submission of articles presenting original and significant research in the aquatic sciences (see [Scope](#)).

Articles that address broad conceptual questions, are interdisciplinary and of wide interest, and that consider further implications and management applications are especially encouraged, given the journal's broad scope. Specialist articles at the forefront of their field are also welcome as long as their context is clearly stated. **Descriptive articles may be considered if they are placed in an appropriate conceptual setting and have global relevance.** However, articles that are purely taxonomic, parochial, describe preliminary or incremental results, or simply present data without context will not be considered.